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Estimating soil carbon fluxes following land-cover change: a test of some critical assumptions for a region in Costa Rica

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Abstract

Changes in soil carbon storage that accompany land-cover change may have significant effects on the global carbon cycle. The objective of this work was to examine how assumptions about preconversion soil C storage and the effects of land-cover change influence estimates of regional soil C storage. We applied three models of land-cover change effects to two maps of preconversion soil C in a 140 000 ha area of northeastern Costa Rica. One preconversion soil C map was generated using values assigned to tropical wet forest from the literature, the second used values obtained from extensive field sampling. The first model of land-cover change effects used values that are typically applied in global assessments, the second and third models used field data but differed in how the data were aggregated (one was based on land-cover transitions and one was based on terrain attributes). Changes in regional soil C storage were estimated for each combination of model and preconversion soil C for three time periods defined by geo-referenced land-cover maps.

The estimated regional soil C under forest vegetation (to 0.3 m) was higher in the map based on field data (10.03 Tg C) than in the map based on literature data (8.90 Tg C), although the range of values derived from propagating estimation errors was large (7.67–12.40 Tg C). Regional soil C storage declined through time due to forest clearing for pasture and crops. Estimated CO_2 fluxes depended more on the model of land-cover change effects than on preconversion soil C. Cumulative soil C losses (1950–1996) under the literature model of land-cover effects exceeded estimates based on field data by factors of 3.8–8.0. In order to better constrain regional and global-scale assessments of carbon fluxes from soils in the tropics, future research should focus on methods for extrapolating regional-scale constraints on soil C dynamics to larger spatial and temporal scales.

Keywords: carbon dioxide fluxes, Costa Rica, land-cover change, regional scale, soil carbon

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Introduction

A major focus of global change research is to understand the cumulative impacts of local land-use changes on global biogeochemical cycles, climatic and hydrologic processes, and land–atmosphere interactions (Wessman, 1992; Vitousek *et al.*, 1997). To assess anthropogenic perturbations on these global systems, we must understand how spatial heterogeneity and non-linear processes hamper efforts to extrapolate from fine to coarse scales (Rastetter *et al.*, 1992; Paustian *et al.*, 1997; Plant, 2000), and how to quantify the errors that accompany extrapolations across scales (Burke, 2000). The conversion of forests to agricultural lands in the

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tropics is a pervasive land-cover change that has consequences for the global carbon cycle (Schlesinger, 1986).

Forest conversion usually results in large fluxes of CO_2 to the atmosphere when biomass is burned, and additional losses from slash and soil organic carbon (soil C) (Houghton, 1995; Melillo *et al*., 1996). Because soils contain more than twice the amount of organic carbon (soil C) found in the terrestrial biota or the atmosphere (Schlesinger, 1997), and approximately one-third of the global soil C pool is in the tropics (Eswaran *et al*., 1993; Jobbágy & Jackson, 2000), there is much concern that land-use change may have significant feedbacks to the global carbon cycle (Batjes & Sombroek, 1997; Post & Kwon, 2000; Amundson, 2001). Losses of soil C also have implications at the local scale. Soil C is an important determinant of site fertility due to its role in maintaining soil physical and chemical properties (e.g. aggregate stability, cation exchange capacity) (Reeves, 1997).

Assessments of the changes in C pools that accompany land-cover change are often limited to specific spatial scales, e.g., global/continental inventories or field studies of several small plots. At the global scale, workers have used simulation or 'bookkeeping' models to estimate CO_2 fluxes that accompany land-use change (Detwiler, 1986; Hall *et al*., 1995; DeFries *et al*., 1999; Houghton, 1999). Bookkeeping methods are based on empirical data to estimate fluxes of CO_2; information on rates of land-cover change is coupled to generalized response functions that describe changes in carbon stored in vegetation, slash, and soil pools following forest clearing (Detwiler, 1986; Houghton, 1995, 1999; Melillo *et al*., 1996). Preconversion soil C pools are estimated from pedon data grouped by a classification scheme such as soil taxonomy or the Holdridge life-zone system. The empirical data that many global inventories (Detwiler, 1986; Houghton, 1999) use to describe changes in soil C stocks following land-use change are derived from key literature reviews (Detwiler, 1986; Schlesinger, 1986; Davidson & Ackerman, 1993). Most global inventories apply a 20–25% reduction in soil C stocks following conversion of forests to pastures in the tropics. Older assessments used a 30–50% reduction following clearing for cultivation (Houghton *et al*., 1983; Detwiler, 1986), while more recent assessments used a 25% reduction following conversion to cropland (Houghton, 1999).

Three components contribute to these assessments: an assumed preconversion distribution of soil C, estimates of land-cover change, and models of the effects of land-cover change on C pools (i.e., response functions). Each of these components has at least two potential sources of error that influence the reliability of these estimates (Table 1). In order to minimize these errors, a large effort has been directed towards improving measurements of deforestation rates (Dale *et al*., 1991). In contrast, few studies have attempted to quantify the error caused by assumptions about preconversion C storage or the effects of forest conversion (see Houghton *et al*., 2000).

Field-scale studies indicate that the response of soil C to forest conversion is more complex than represented by current global carbon accounting methods (Guo & Gifford, 2002; Murty *et al*., 2002). For example, in a recent review of paired forest-to-pasture comparisons, Veldkamp (in press) found no consistent change in tropical soil C stocks expressed on a common mass basis (i.e. corrected for the effects of compaction). On average, soil C stocks *increased* by 5% (to 20 cm depth), but the changes ranged from losses of 63.4% to gains of 62.3%. Veldkamp notes that the average percent change in soil C stocks should not be applied in global inventories because the field studies do not represent the area-weighted distribution of edaphic characteristics and management practices in the tropics. Many factors may explain the large variation in the response of soil C pools to land-cover change, including vegetation productivity and rooting patterns (Fisher *et al*., 1994), management practices (Trumbore *et al*., 1995; Neill *et al*., 1997; Fearnside & Barbosa, 1998), precipitation (Jackson *et al*., 2002), soil age and degree of weathering (Allen, 1985), topography (Powers and Veldkamp, unpublished results), or soil texture (Veldkamp, in press). Regardless of the source of error, the discrepancy between plot-level field studies and the 20–40% losses that are often used in global inventories suggests that our knowledge of change in soil C following clearing does not adequately represent the high diversity of soils in the tropics (Richter & Babbar, 1991).

To make reliable global-scale estimates, we need to understand the role of site-specific factors and spatial heterogeneity in mediating the response of the soil C pool to land-cover change, and how to extrapolate results from fine to coarse scales. The goals of this study were to (i) examine the effects of assumptions of preconversion soil C storage and land-cover change on estimates of soil C storage over time, and (ii) to identify avenues for further work to reduce uncertainty surrounding these estimates. We contrasted a 'top-down' approach (*sensu* Wessman, 1992) to soil C budgets used in global/continental inventories (i.e., applying generalized response functions to patterns of land-cover change) with a 'bottom-up' approach based upon extrapolating field data to the landscape scale. We compared these methods in a region in northeastern Costa Rica as a case study because of the availability of multitemporal land-cover maps and extensive measurements of soil C stocks under different land uses.

Table 1 Sources of error/uncertainty in global and regional soil C inventories

Component of soil C budget	Source of error/uncertainty	Global inventories	This study
Land-use change statistics from forestry and agricultural surveys	Difficulty in estimating selectively logged areas; unreliable surveys	Use high and low estimates of deforestation rates from different sources	Not applicable
Land-use change statistics from remote sensing	Classification accuracy of remotely sensed data; spatial resolution of the sensor	Not addressed	Quantified by κ statistic; error not propagated in this study
	Temporal resolution of time-series	Choose one baseline year or decade	Not addressed
Preconversion soil C estimates from life-zone classification	Measurement error (e.g. errors in soil pedon data base, no reliable climate data for mapping Holdridge life zones)	Not addressed	Not addressed
	Aggregation error (delineating map units, etc.)	Not addressed	Not addressed
Preconversion soil C estimates from terrain analysis	Errors and poor resolution of DEMs and map layers derived from DEMs	Not applicable	Not addressed
	Measurement error of soil C stocks (within-field variability)	Not applicable	Compared with independent data set; laboratory errors are minimal
	Estimation error	Not applicable	Propagate using high and low estimates of coefficients from regression equations
	Model error	Not applicable	Not addressed
Effects of land-use change on soil C storage estimated from averaged data	Measurement error in empirical studies	Not addressed	Compared with independent data set; laboratory errors are minimal; site selection may bias estimates
	Aggregation error (landscape-scale heterogeneity)	Not addressed	Compared with estimates from geographically explicit model
Effects of land-use change on soil C storage estimated from linear regression and terrain attributes	Errors and poor resolution of DEMs and map layers derived from DEMs	Not applicable	Not addressed
	Measurement error of soil C stocks	Not addressed	Compared with independent data set; laboratory errors are minimal
	Estimation error	Not addressed	Propagate using high and low estimates of coefficients from regression equations
	Model error	Not addressed	Not addressed

DEM, digital elevation model.

Materials and methods

Approach

We built a database of raster data layers including land-cover maps, elevation, and percent slope in Arc/Info GRID for a 140 000 ha area of northeastern Costa Rica. We used both literature and field data to construct two maps of preconversion soil C (Table 2). For each predisturbance soil C map, we applied three models of the effects of land-cover change on soil C storage to calculate the change in regional soil C for three time periods defined by geo-referenced land-cover maps.

Many sources of error and uncertainty accompany large-scale soil C budgets (Phillips *et al.*, 2000) (Table 1). They include errors of measurement due to sampling

Table 2 Methods used to generate two predisturbance soil C maps and three models of land-cover change effects

Map/model	Description and data sources	Reference
Pre-conversion soil C maps		
Life-zone map	Uses values applied in global/continental inventories to simulate forest soil C maps in the wet tropical forest life zone. 71 Mg C ha^{-1} assigned to all grid cells	Detwiler (1986)
Field data map	Measured C stocks in forest plots from the region extrapolated to each grid cell in the landscape with linear regression and data layers of elevation and percent slope	Powers & Schlesinger (2002)
Land-cover change effects models		
Literature	Simulates the changes in soil C stocks with land-cover changes that are used in global/continental inventories	Detwiler (1986)
Land cover	Uses the average, measured values for changes in soil C stocks grouped by land-cover transition	Powers (2001), Powers (in press)
Terrain	Uses regression equations derived from field data to calculate soil C losses as a function of elevation and % slope when forest grid cells change to pasture. Other land-cover transitions are implemented as in the 'land cover' model	Powers (2001)

(image classification accuracy, laboratory analyses, etc.), estimation (a function of both sample size and variance), and modeling errors (selection of the appropriate functional form of the regression). Here, we quantified two sources of estimation error that occur when propagating uncertainty in field data to the landscape scale: preconversion soil C maps, and field-based estimates of the effects of land-cover change on soil C storage. Our objective was to determine research priorities to reduce uncertainty. Thus we examined preconversion soil C maps and land-cover change effects as two separate sources of uncertainty, but we did not quantify their joint effect.

Study area

The 140 000 ha region is located in northeastern Costa Rica, in the canton of Sarapiquí (the northwestern corner is 10°34′27.84″N, 84°13′56.86″W; the southeastern corner is 10°14′41.47″N, 83°52′12.17″W). The region includes a long elevation gradient (~40–800 m above sea level (a.s.l.)), on the northern flanks Volcan Barva and Volcan Congo. Mean annual temperatures vary from ~25 °C in the lowlands to 21 °C at high elevations. The mean annual precipitation across the region is ~4065 mm (standard deviation = 415 mm), with no month receiving less than 100 mm of rain (Powers & Schlesinger, 2002). Soil parent materials are volcanic, and soils differ mainly in age and geomorphic origins. High-elevation soils mapped as Inceptisols or Andisols are less weathered than low-elevation Ultisols and Inceptisols (Grieve *et al*., 1990, Powers & Schlesinger, 2002). Potential vegetation is mapped as tropical wet forest, tropical wet forest/cool transition, and tropical premontane rain forest according to the Holdridge life-zone system, but the entire study area would be considered tropical wet forest according to the aggregated vegetation classifications used in global inventories (e.g. Detwiler, 1986; Brown & Lugo, 1990; Houghton, 1999). A digital elevation model (92 m grid DEM) encompassing the entire study region was made available by the National Intelligence Mapping Agency through a cooperative agreement with NASA's Goddard Space Flight Center.

Spatial and temporal patterns of land-cover change

Although paleo-ecological studies indicate the pre-Columbian presence of people in the region, Sarapiquí was largely forested before the arrival of modern agriculture (Pierce, 1992; Montagnini, 1994; Kennedy & Horn, 1997; Read *et al*., 2001). Much of the recent land-use and land-cover history of Sarapiquí has been documented and is summarized here briefly (Pierce, 1992; Butterfield, 1994; Montagnini, 1994; Read *et al*., 2001). Clearing of forests for cattle pastures began in the late 1950s and accelerated through the 1960s and 1970s. Falling prices for beef exports combined with the designation of much of the remaining forested lands as a Protected Zone adjacent to Braulio Carrillo National Park led to a decline in deforestation rates in the 1980s (Lehmann, 1992; Read *et al*., 2001). Predominant land-cover transitions since the mid-1980s include the conversion of pastures to intensively managed cash crops (banana (*Musa acuminate*) and pineapple (*Ananas comosus*)) and the regeneration of secondary forests on abandoned pasture land. Shifting cultivation is not widespread in this region.

The spatial and temporal patterns of land-cover change in the study region have been documented in a chronosequence of maps derived from multispectral scanner (MSS) and thematic mapper (TM) satellite images (Powers, 2001; Read *et al*., 2001). Details of the classification procedures and κ statistics (for accuracy assessment) can be found in Read *et al*. (2001) and Powers (2001). Land-cover categories for all maps (1976, 1986, and 1996) were aggregated into the following cover types: forest (primary, secondary, and scrub), pasture, crops/bare soil, water (rivers/lakes), and clouds. The minimum mapping unit for all land-cover maps was 3 ha. For the analyses described below, the three land-cover maps were resampled to a common spatial resolution (92 m cell size) to match the digital elevation model. The temporal resolution of the land-cover maps ($\sim$10 years intervals) cannot resolve the dynamics of patches that were cleared, abandoned, and regenerated to forests within these intervals. However, the series captures the net land-cover trajectories, and the spatial patterns appear consistent between time steps (Fig. 1).

Two preconversion (1950) soil C maps

For both preconversion soil C maps, we assumed that the only land-cover types in 1950 were forest and rivers. Although 1950 is an arbitrary year, an extensive set of aerial photos taken in 1952 (archived at the La Selva Biological Station) suggests that this is a reasonable

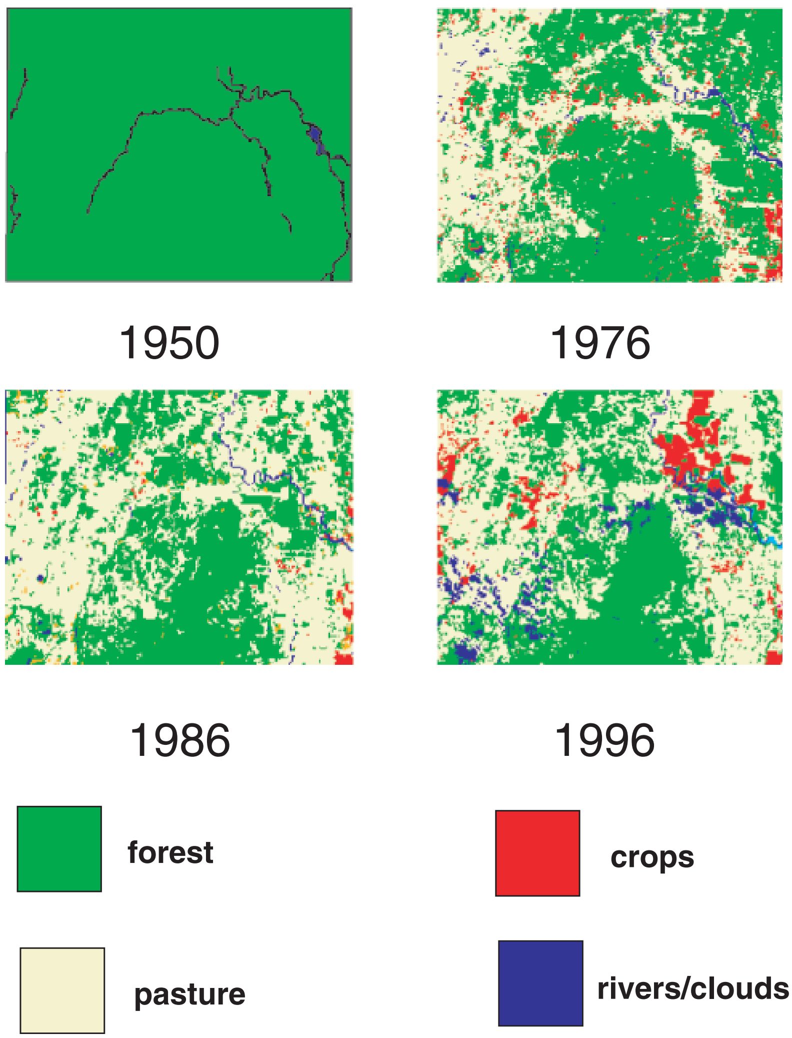

Fig. 1 Spatial patterns of land cover for four time periods. In 1950, the only two land covers are assumed to be forest and rivers. The study area is $\sim 35 \times 40\,\mathrm{km}^2$.

assumption. We also assumed that the forest soil C pools were in a steady state prior to land-cover change and that changes in regional soil C storage are the result of land-cover change alone. For all of the subsequent analyses, regional soil C storage was calculated after eliminating areas covered by clouds ($\sim$4.5% of the 1996 land-cover map), rivers, and portions of the study region $>$800 m a.s.l. in elevation, the maximum extent of the field data.

The life-zone map of soil C was generated from literature values presented in Detwiler (1986). The entire study region would be classified as tropical wet forest in his analyses. Detwiler (1986) assigned these forest soils an average value of 150 $Mg\,C\,ha^{-1}$ to a depth of 1 m and 81 $Mg\,C\,ha^{-1}$ to a depth of 0.4 m. Because all of the empirical data reported here are for a soil depth of 0.3 m, we assigned forest soils a value of 71 $Mg\,C\,ha^{-1}$. This value is $>$75% of Detwiler's 0.4 m value because soil C concentrations decline with depth. It represents an average of 81 and 60.75 $Mg\,C\,ha^{-1}$, or 75% of Detwiler's 0.4 m value. Carbon stocks in units of $Mg\,C\,ha^{-1}$ were converted to a grid cell-sized area (i.e. 8464 m^2). Although the Detwiler values are presented without error, the variability in soil C for wet tropical forests reported by Post *et al*. (1982) can be used to estimate a standard error around the mean value of 71 Mg C of 8.5.

The field-data soil C map was created from measurements of soil C in 35 forest plots located throughout the region (Powers & Schlesinger, 2002). Soil C storage increases linearly with elevation, but the rate of increase varies between low-elevation ($<$120 m a.s.l.) and high-elevation soils ($>$120 m a.s.l.). Elevation is a proxy for temperature and clay mineralogy, both of which influence forest soil C pools (Oades, 1988). In the low-elevation soils, percent slope (topographic relief) also explains a significant fraction of the variation in soil C storage. Using the DEM and a slope map derived from the DEM, the study area was divided into zones above and below 120 m a.s.l., and soil C storage (expressed on a per-grid cell basis) was assigned to each grid cell based upon linear regression equations presented in Table 3. The estimation errors associated with the linear regression equations were propagated for the region by constructing two additional soil C maps that represented high and low estimates of soil C storage. These maps were created from the regression equations presented in Table 3, but with the intercepts and coefficients for elevation and percent slope increased or decreased by one standard error.

Three models of soil C changes following land-cover change

We applied three models of land-cover change effects, which we refer to as the 'literature', 'land-cover', and 'terrain' models (model definitions are presented in Table 2). All three models of the changes in soil C storage following land-cover change were implemented at discrete time steps. Our implicit assumption is that if the land cover of a grid cell changes during a time step, all of the changes in soil C stocks occur during that time step. This approach contrasts with the time-integrated response functions used by Detwiler and Houghton (Houghton *et al*., 1983; Detwiler, 1986; Houghton, 1999). However, we believe that this assumption is justified for this region because the changes in soil C pools following the dominant land-cover transitions, deforestation, and pasture to crop conversions occur rapidly (usually $<$3 years) (Powers, in press). In this region, differences in soil C stocks as a function of land use are correlated with vegetation productivity, soil mineralogy, and topography, and not with time since conversion (Powers, in press; Powers and Veldkamp, unpublished results).

Many of the parameter values for changes in soil C storage in the literature model were taken from Detwiler (1986) (Table 4). He assigned a 20% reduction in soil C storage following the clearing of forest for pastures and a 40% reduction following clearing for

Table 3 Linear regression equations that relate field-measured soil C storage under forest vegetation to terrain attributes

Elevation zone	Model terms	Coefficients (standard error)	Coefficient (P-value)	df	Multiple r^2	Overall P-value
$<$120 m	Intercept	35.02 (6.95)	$<$0.0001	23	0.54	0.0001
	Elevation	0.53 (0.10)	$<$0.0001			
	Percent slope	−0.33 (0.10)	0.004			
$>$120 m	Intercept	47.10 (10.83)	0.003	7	0.49	0.03
	Elevation	0.06 (0.02)	0.03			

Table 4 Parameters of changes in percent soil C storage due to land-cover change for the three models

Land-cover transition	Literature model	Land-cover model	Terrain model
Forest → forest	0	0	0
Forest → pasture	−20	−1.64 (−4.10, + 0.82)	Regression equations given in (Table 5)
Forest → crop	−40	−6.39 (−11.39, −1.39)	−6.39 (−11.39, −1.39)
Pasture → forest	+ 20	−6.01 (−0.66, + 5.35)	−6.01 (−0.66, + 5.35)
Pasture → pasture	0	0	0
Pasture → crop	0		
Banana		−9.17 (−14.58, −3.77)	−9.17 (−14.58, −3.77)
Pineapple		−16.55 (−21.47, −11.64)	−16.55 (−21.47, −11.64)
Crop → forest	+ 40	0	0
Crop → pasture	0	0	0
Crop → crop	0	0	0

cultivated fields (Detwiler, 1986). The value for conversion of forest to crop is similar to those from two recent literature surveys, which report soil C losses from 22% to 42% following conversion of forest to cultivated crops worldwide (Guo & Gifford, 2002; Murty *et al.*, 2002), but higher than the values based on field data for this region (Table 4). He also assumed that soil C storage following abandonment of managed lands returns to preconversion values after 35 years. As a simplifying assumption, we assigned the initial soil C storage to patches of managed lands that had been allowed to regenerate to forest (Table 4). Detwiler did not incorporate model terms for the intensification of agriculture, i.e. the conversion of pasture to cultivated fields. Accordingly, we assumed no changes in soil C storage following conversion of pastures to crops in the global model. Because the Detwiler values were reported without error, uncertainty terms are not included.

Changes in soil C stocks under different land-cover transitions for the two models based on field data are presented in Table 4. The land-cover and terrain models used the same values for forest to crop, pasture to forest, and pasture to crop land-cover transitions; the conversion of forest to pasture was parameterized differently in the two field data models (described below).

Cropped fields are rarely converted to forests or pastures in this region, and the effects of this land-cover trajectory on soil C storage were not measured.

The effects of pasture establishment were estimated from 43 forest-to-pasture comparisons (Powers, 2001). In this region, soil C storage may increase or decrease following conversion of forests to pastures. The average effect of the conversion of forests to pastures across the region was a small decrease in soil C storage of 1.6% (nonsignificant). This value was assigned to all forest-to-pasture conversions in the land-cover model (Table 4),

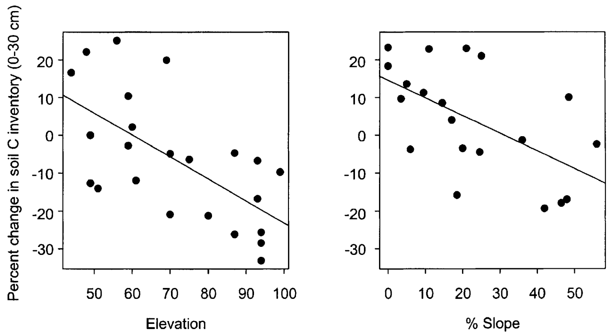

Fig. 2 Percent change in soil C storage (0–30 cm) following forest clearing and pasture establishment in (a) low- (<100 m), and (b) high-elevation (>100 m) soils. Regression equations for the solid lines are represented in Table 5.

and compares favorably with recent estimates from a global literature review and a meta-analysis that found increases in soil C storage from 6.4% to 8% worldwide (Murty *et al.*, 2002). The 'terrain' model used linear regression to estimate the response of the soil C pool to pasture establishment, which depended upon elevation (Fig. 2, Table 5). In sites below 100 m a.s.l., the percent change in soil C storage is negatively related to elevation (Fig. 2). In sites above 100 m a.s.l., the percent change in soil C storage is negatively related to percent slope.

Estimating regional soil C storage

Regional soil C storage was estimated for each combination of preconversion soil C map and model of the effects of land-cover change. For each time step, the previous distribution of land-cover types in the region was compared with the new distribution; new values of soil C were assigned to grid cells that had changed cover types according to the rules outlined in

Table 5 Linear regression equations expressing the percent change in soil C storage with the conversion of forest to pastures to terrain attributes under the 'terrain model' of land-cover change effects (data are from Powers, 2001)

Elevation zone	Model terms	Coefficients (standard error)	Coefficient (P-value)	df	Multiple r^2	Overall P-value
<100 m	Intercept	34.87 (11.29)	0.0056	21	0.40	0.001
	Elevation	−0.58 (0.15)	0.0011			
>100 m	Intercept	14.61 (4.45)	0.0041	18	0.33	0.008
	percent slope	−0.47 (0.16)	0.0075			

Table 4. At each time step, the total regional soil C storage to 0.3 m depth was calculated in Tg C (i.e. 10^{12} g C) by summing the values of each grid cell over the entire region. For land-cover change models based on field data, we derived confidence intervals that reflected estimation error for regional soil C storage at each time step. Parameters describing the changes in soil C storage were decreased or increased by one standard error of the mean, to create maps of the low and high estimates of soil C storage.

Results

Spatial and temporal patterns of land-cover change

Summary statistics for the percentages of land undergoing different land-cover transitions show that the establishment of cattle pastures has dominated land-use change for much of the recent history of Sarapiquí (Table 6). Most of this conversion occurred between 1950 and 1976 (Table 6); however, pasture establishment remained the dominant land-cover change until 1986. In contrast, between 1986 and 1996 the loss of forests for pastures was approximately balanced by regeneration of forests on previously grazed lands. In addition, ~5.5% of the old pastures was converted to crops.

Changes in regional soil C storage

The total regional soil C storage before land-cover change to a depth of 0.3 m based on the life-zone data was 8.90 Tg C (the standard error estimated from data in Post *et al*., 1982 is 1.1 Tg C). This value was close to the total soil C storage estimated from field data, 10.03 Tg C, and was well within the lower and higher estimates from the field-data soil C map of 7.67 and 12.40 Tg C (Table 7). Regional soil C storage decreased through time under all models of land-cover effects and preconversion soil C, but the final estimated C storage was much less under the literature model (Table 7). Soil C storage from 1976 to 1996 using the land-cover model was significantly less than initial soil C from the life-zone predisturbance soil C map. In contrast, the confidence intervals for the terrain model were large, and the initial value of total soil C storage from the life-zone map fell within the lower and higher estimates of soil C storage in 1976, 1986, and 1996.

Table 6 Percent of land undergoing different land-cover transitions in the study region for three time periods, 1950–1976, 1976–1986, and 1986–1996

Land-cover transition	1950–1976	1976–1986	1986–1996
Forest → forest	58.1	38.5	34.6
Forest → pasture	36.7	19.6	7.9
Forest → crop	5.2	0.2	1.3
Pasture → forest		3.9	7.1
Pasture → pasture		32.2	42.5
Pasture → crop		0.4	5.6
Crop → forest		1.4	0.2
Crop → pasture		3.3	0.3
Crop → crop		0.4	0.5

Areas remaining in forest, pasture, or crops for the duration of a given time period are designated as forest → forest, pasture → pasture, and crop → crop, respectively. Rivers and clouds were excluded from these summaries, thus the values within a column sum to 100%.

All of the model estimates of regional C storage in 1976, 1986, and 1996 based on the field-data soil C map fell within the large confidence intervals that surrounded the initial, preconversion soil C estimate (Table 7). By 1996, the total regional soil C storage was 86.5%, 96.4%, and 98.2% of the 1950 field data value, for the literature, land-cover, and terrain models, respectively (Fig. 3a).

Assuming that all loss of soil C occurred through microbial respiration and not erosion, the fluxes of CO_2 from the soil to the atmosphere can be estimated for the three time periods: 1950–1976, 1976–1986, and 1986–1996 (Fig. 3b). The estimated CO_2 fluxes from 1950 to 1976 are ~3.8–8.0 times greater under the literature model than under the two models based on field data (Fig. 3b). The estimated CO_2 fluxes from 1976 to 1986 were much less than the fluxes from the previous time period, reflecting the decrease in deforestation rate

Table 7 Total regional soil C storage (Tg C) to 0.3 m, estimated from different combinations of preconversion soil C maps and models of the effects of land-cover change

	Literature model	Land-cover model	Terrain model
Life zone soil C map			
1950	8.90 (7.8, 10.00)	8.90 (7.80, 10.00)	8.90 (7.8, 10.00)
1976	8.01	8.75 (8.62, 8.83)	8.79 (8.37, 9.21)
1986	7.74	8.66 (8.61, 8.73)	8.77 (8.53, 9.08)
1996	7.69	8.58 (8.55, 8.69)	8.75 (8.65, 8.96)
Field-data soil C map			
1950	10.03 (7.67, 12.40)	10.03 (7.67, 12.40)	10.03 (7.67, 12.40)
1976	9.02	9.86 (9.75, 9.98)	9.89 (9.42, 10.36)
1986	8.72	9.76 (9.73, 9.85)	9.87 (9.60, 10.21)
1996	8.68	9.67 (9.66, 9.81)	9.85 (9.74, 10.08)

Parentheses indicate low and high estimates from ± 1 SE of each model coefficient. The high and low estimates for regional soil C storage in 1950 reflect uncertainty from the initial soil C map, and high and low estimates for dates 1976–1996 reflect uncertainty in the model of land-use change effects.

(Table 6); the literature model still yielded the highest flux estimates. However, from 1986 to 1996, the land-cover model produced the highest estimated CO_2 fluxes.

Discussion

Our two major findings were that the uncertainty from estimation error of predisturbance soil C stocks from both life-zone and field data was large relative to that associated with land-use effects, and calculated regional CO_2 fluxes varied dramatically depending on the assumed effects of land-cover change on soil C pools. Both the patterns of land cover and field data on soil C stocks are specific to this volcanic region, and should not be generalized to other areas. Moreover, our flux estimates rely on the unsubstantiated assumption of no erosion, which implies that all soil C losses result from microbial respiration (i.e. CO_2 effluxes). Some hydrologic studies have shown that erosion may influence all vegetation cover types (including forests) in this landscape (Jansson, 2002), thus, our flux estimates represent upper boundaries. Despite these caveats, this case study has implications for soil C assessments in different regions and at larger spatial scales.

Because of the inconsistent responses of soil C pools to land-cover change and the larger fluxes from biomass burning, some authors have suggested omitting the changes in soil C in regional/continental inventories of carbon balances (Houghton *et al*., 2000). However, in an inventory that accounted for soil C dynamics under both productive and degraded pastures in Brazil, Fearnside & Barbosa (1998) found that carbon fluxes from the soil pool were $\sim$20% of Brazilian fossil fuel emissions for 1990. To better constrain carbon fluxes from soil C pools and determine when and where regional or global assessments should include these terms, the question becomes where to direct future research. We focus first on differences in estimates derived from the two predisturbance soil C maps and second on the differences from models of land-use change effects.

Our ability to assess the effects of changes in land use on biogeochemical cycles depends upon reliable estimates of ecosystem properties (e.g. soil C storage) and processes before land-cover change (DeFries *et al*., 1999). We often rely on models or extrapolations of field-scale data under 'natural vegetation' to reconstruct past ecosystem conditions, because many terrestrial ecosystems are already subject to human uses and archived data are rarely available at the necessary spatial scales. In this region, the highly aggregated soil C estimates from life-zone data were not substantially different from estimates based upon extensive field data, suggesting that further work may not refine these estimates. For example, the coefficient of variation for soil C stocks (0.3 m depth) in 35 rain forest plots located across an 800 m elevation gradient in the study region was 20%, which is relatively small (Powers & Schlesinger, 2002). On regional and global scales, soil C may be less variable than other ecosystem properties, such as aboveground biomass. For example, Houghton (1999) reported an 83-fold variation for C stored in vegetation among biomes, compared with a five-fold variation for soil C.

There were large differences in estimated regional soil C storage between the literature model of land-use effects and the two models based on field data. Because the predominant land-cover transition was the conversion of forest to pasture, the differences between model estimates of CO_2 fluxes largely reflect assumptions

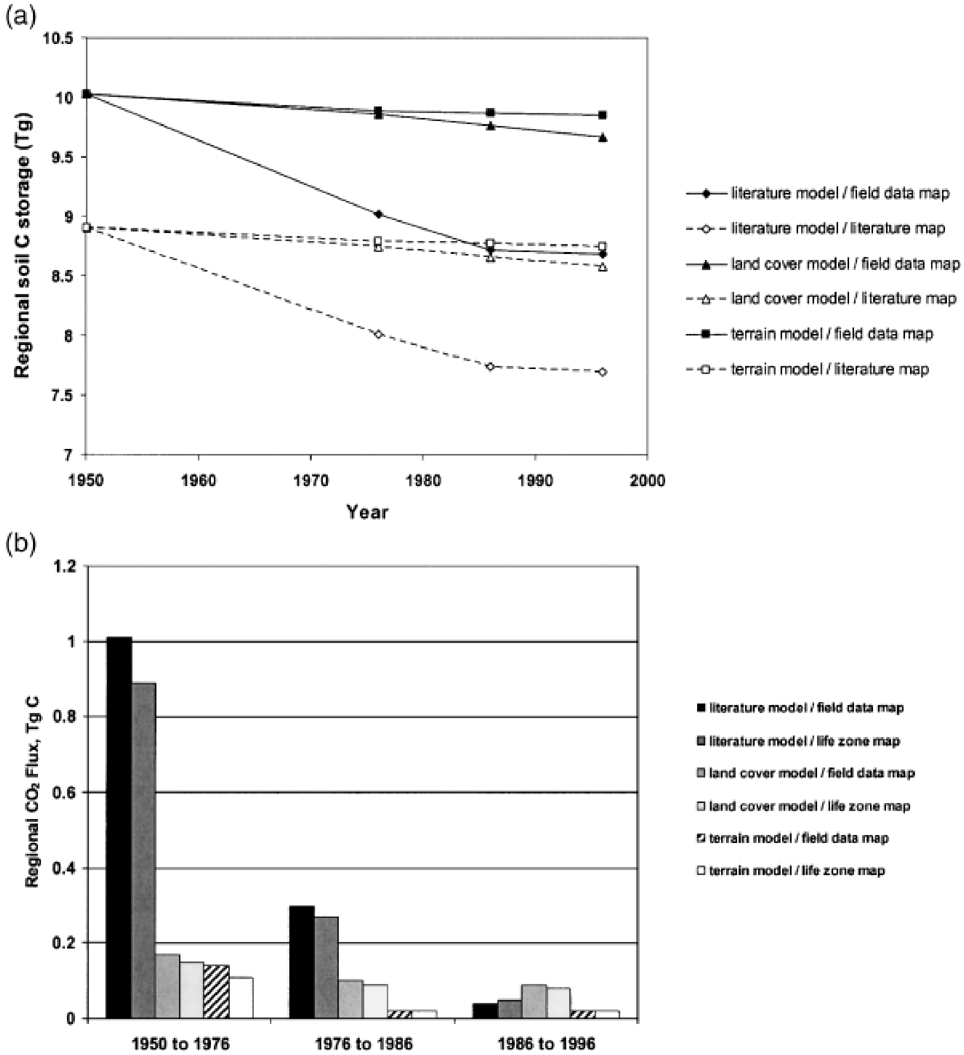

Fig. 3 (a) Total, regional soil carbon storage (0–30 cm) based on two different initial soil C maps and three different models of land-cover change, and (b) estimated regional CO_2 fluxes (Tg C) for specific time periods under different assumed effects of land-cover change and initial, predisturbance soil C storage. In both figures, the first legend entry represents the model of land-cover effect (literature, land cover, and terrain models) and the second legend entry represents predisturbance soil C map (field data or life-zone maps).

about the effects of pasture establishment on soil C pools. On average, the measured losses in soil C storage with conversion of forests to pastures were modest compared with the 20% reduction in soil C storage (0.3 m depth) that we assumed in the literature model. This value is actually conservative compared with the 25% loss in soil C storage following conversion of undisturbed tropical equatorial forest in South America used in a recent study (Houghton, 1999), but larger than more recent estimates from literature reviews (Guo & Gifford, 2002; Murty *et al*., 2002).

However, the average percent change in soil C stocks across this region obscures a large range of variation. Depending upon topography and soil mineralogy, soil C stocks in pastures change from −25% to + 25% of the initial forest inventories (Powers and Veldkamp, unpublished results). Thus, whether pasture soils are C sources or sinks in this region depends upon the area-weighted distribution of soil-forming factors such as topography. If the study landscape was comprised entirely of landform units that lose C following conversion to pastures (e.g. steep slopes), we would expect small differences between the terrain and literature model estimates of CO_2 fluxes. The modest differences between the land-cover and the terrain models reflect this landscape heterogeneity, because the field sites were not selected based upon area-weighted sampling of different slope positions and elevations in the landscape. In conclusion, because the variable response of soil C pools to land-cover change can be explained in terms of site-specific variation in soil-forming factors, vegetation productivity, etc., a promising direction for future studies will be to develop methods for extrapolating regional-scale constraints on soil C dynamics to larger spatial and temporal scales.

Implications for soil C assessments

The 'top-town' approach employed in global and continental inventories of CO_2 fluxes due to tropical deforestation necessitates averaging over spatial heterogeneity in both initial soil C distributions and the effects of land-cover change. When compared with a 'bottom-up' approach based on field data at a common regional scale, estimated CO_2 fluxes using the literature model differed considerably. In this study, estimation error was large for both initial preconversion soil C maps and regional soil C storage given land-use change. If all potential sources of error and uncertainty had been considered, any signal of land-use change on soil C storage likely would be undetectable in this landscape. Thus, regional and global assessments that account for only uncertainty in land-use change estimates must be viewed cautiously. The development of well-constrained, global estimates of the CO_2 fluxes that accompany tropical land-use change will require a better understanding of local and regional variations in soil C pools, how land-use change influences soil C storage, and statistical techniques to account for all potential sources of uncertainty.

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